INDIVIDUAL THYROID DOSE ESTIMATES FOR A CASE-CONTROL STUDY OF CHERNOBYL-RELATED THYROID CANCER AMONG CHILDREN OF BELARUS—PART II. CONTRIBUTIONS FROM LONG-LIVED RADIONUCLIDES AND EXTERNAL RADIATION

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Abstract—**Significant quantities of long-lived radionuclides were released to the environment during the Chernobyl nuclear power plant accident in 1986. These radionuclides contributed to radiation doses due to ingestion of contaminated foods and external exposure from the ground deposition that resulted. The contributions of these exposure pathways to thyroid doses received by subjects of an epidemiologic study of children from Belarus are evaluated and presented. The analysis shows that ingestion of the long-lived radionuclides, primarily radiocesium, typically contributed a small percentage of the total thyroid dose received by the study subjects. The median and mean fractional contributions were 0.76 and 0.95%, respectively. The contribution of external exposure to the thyroid dose was generally larger and more variable, with median and mean contributions of 1.2 and 1.8% of the total thyroid doses, respectively. For regions close to the reactor site, where radionuclide deposition was highest, the contributions of radiocesium ingestion and external exposure were generally lower than those of the short-lived radioiodine isotopes (^{132}I and ^{133}I) and their precursors (^{132}Te). In other areas, the contributions of these two pathways were comparable to those of the short-lived radioiodines. For all subjects, intakes of ^{131}I were the primary source of dose to the thyroid.**
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Key words: Chernobyl; thyroid; exposure, population; children

INTRODUCTION

A TEAM of Belarusian, Russian, and American scientists has investigated whether thyroid cancers in children in Belarus were related to radiation exposure following the accident at the Chernobyl Nuclear Power Plant (NPP) in April 1986. Clinical and epidemiological aspects of the case-control study have been published (Astakhova et al. 1998). A companion paper (Gavrilin et al. 2004) addresses doses to the thyroids of cases and controls that were due to exposure to ^{131}I and to shorter-lived radioiodines, primarily ^{132}I and ^{133}I. This paper considers (1) thyroid doses due to ingestion of radiocesium, primarily ^{137}Cs (the most important long-lived contributor) and other isotopes that were present in foods soon after the accident and (2) thyroid doses due to external exposure to contaminated surfaces as the result of deposition of those radionuclides on the territory of Belarus.

External exposure of the population after the Chernobyl accident was mainly due to deposition of gamma-emitting radionuclides, including ^{95}Zr-Nb, ^{131}I, ^{132}Te-I, ^{134}Cs, ^{137}Cs, and ^{140}Ba-La. Because ^{137}Cs was most commonly measured throughout the contaminated zones, deposition densities of the other radionuclides have generally been related to that for ^{137}Cs. In this paper, external exposures from ^{137}Cs and the other radionuclides are calculated for the 30-km zone and nine other defined geographic areas where study subjects resided in Belarus.

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Initially, the surfaces of green vegetables and herbs were contaminated directly and cows' milk and meat were contaminated indirectly as a result of the fallout. Consumption of these foodstuffs was the main pathway for internal exposure soon after the accident. The contribution from inhalation of radiocesiums and other radionuclides in the first few weeks following the accident was negligible compared to exposure from food consumption (UNSCEAR 1988), even for the population in southern Belarus that was close to the Chernobyl NPP (Knatko et al. 1993).

During the first few weeks after the accident, the concentrations of ^{137}Cs in milk were due to direct fallout on vegetation that decreased with a half-time of about 15 d as ^{137}Cs was lost from the vegetation and transferred to the ground. Later on, ^{137}Cs contamination of vegetation resulted from root uptake and decreased much more slowly, with half-times of the order of years, according to the dynamics of cesium in the agroecosystem (Alexakhin et al. 1996). Locally produced foodstuffs reflected this increase and were the main source of internal exposure to long-lived radionuclides for inhabitants of Belarus. The contributions to internal exposure of the thyroid from ingestion of ^{134}Cs and ^{137}Cs are considered in this paper.

The methods for ingestion dose calculation are described next. The methods for external dose calculation and results of both sets of calculations follow. The contributions of external exposure and ingestion of long-lived nuclides to the total thyroid doses for cases and controls are then discussed.

METHODS FOR INGESTION DOSE CALCULATIONS

After the decay of ^{131}I and shorter-lived radioiodines, the long-lived radionuclides ^{134}Cs and ^{137}Cs were the main contributors to the thyroid dose as the result of ingestion of contaminated foods. The contribution of ^{136}Cs (13.2-d half life) to the total ingestion dose from radiocesium during the first year after the accident was estimated to be less than 1% (Minenko et al. 1996). These radionuclides are readily detectable using whole-body counting (WBC) techniques. Many WBC measurements have been made in Belarus since the Chernobyl accident, but the WBC program was not designed with an epidemiologic study in mind. While many cases of thyroid cancer are from regions where there were active WBC programs, the controls selected by the epidemiologists were from all parts of Belarus including areas where few WBC measurements were made. Individual WBC results, even had they been available and retrievable for all cases and controls, would likely not have provided an adequate time history of the subjects' body burdens. The large volume of WBC data for residents of Belarus was collected over many years, which span the period of interest for the epidemiologic study (Drozdovitch and Minenko 1996). This body of data is a valuable resource that can be used to estimate doses to both cases and controls in a consistent manner. The dosimetric model is based on computed intakes and measured radiocesium body burdens of groups of children whose ages were comparable to the subjects of the study.

Internal thyroid dose from ingestion of radiocesium to the members of k^{th} age group constantly living on the contaminated territory can be calculated as a sum of doses from ^{134}Cs and ^{137}Cs:

$$^{int}D_k = \sigma_{137} \sum_i DF_{k,i} \int_{t_1}^{t_2} I_{k,i}(t)dt \qquad (1)$$

where

$^{int}D_k$ = absorbed dose to the thyroids of members of k^{th} age group from ingestion of radionuclides, Gy;

σ_{137} = ground deposition density of ^{137}Cs, kBq m^{-2};

$DF_{k,i}$ = age-dependent dose coefficient for thyroid for ingestion of the i^{th} radionuclide, Gy Bq^{-1};

$I_{k,i}(t)$ = intake function, normalized to ^{137}Cs deposition density, of the i^{th} nuclide by the members of the k^{th} age group, Bq d^{-1} per kBq m^{-2}; and

t_1, t_2 = times (d) corresponding to the beginning of activity ingestion and the case-control study reference date (31 December 1991), respectively.

Data on the deposition density of ^{137}Cs were obtained from the State Committee on Hydrometeorology of Belarus (SCHM 1994). Dose coefficients for particular ages were estimated by interpolation of values for ages (infant, 1 y, 5 y, 10 y, 15 y, and adult) given in ICRP Publication 56 (ICRP 1989). Further discussion of the intake functions and of the resultant body burdens under various conditions is presented below.

Estimation of radionuclide intakes and doses during 1986

Deposition of radionuclides onto pastures and other vegetation occurred soon after the accident. Edible herbs and green vegetables were contaminated directly, and indirect contamination of milk and meat also occurred. The companion paper (Gavrilin et al. 2004) discusses the milk pathway for ^{131}I in detail. The milk pathway for other radionuclides, particularly ^{134}Cs and ^{137}Cs, is similar. The time history of the initial radionuclide ingestion was dependent upon the retention of deposited radionuclides by forage.

It was shown by Savkin et al. (1996) that an adequate description of the time history of ^{137}Cs milk contamination during the first months after the accident could be obtained using a modification of the model originally proposed by Garner (1967). The milk concentration is related to the ^{137}Cs deposition density, expressed in kBq m^{-2}, at the location of interest. The mathematical expression for the modified empirical model is:

$$C_m(t) = C_0 \sigma_{137,nv}^{-0.5} \sum_{j=1}^{6} a_j e^{-\lambda_j t} \qquad (2)$$

where

$C_m(t)$ = normalized concentration [(Bq L^{-1}) per (kBq m^{-2})] of ^{137}Cs in milk;
C_0 = empirical coefficient with units of (Bq L^{-1}) per (kBq m^{-2});
$\sigma_{137,nv}$ = numerical value of σ_{137}, the ground deposition density of ^{137}Cs (kBq m^{-2}); and
a_j, λ_j = parameters of the modified Garner model, shown in Table 1.

The intake of radiocesium in the entire diet may be estimated using information on the intake of radiocesium in milk. Both the milk concentration and the intake are normalized to the ^{137}Cs deposition density. The following equation shows the relationship:

$$I_k = K\nu_k C_m(t) = K\nu_k C_0 \sigma_{137,nv}^{-0.5} \sum_{j=1}^{6} a_j e^{-\lambda_j t} \qquad (3)$$

where K = a parameter that reflects the difference between the entire radiocesium intake and the intake from milk alone; and ν_k = milk consumption rate (L d^{-1}) of the k^{th} age group.

To validate the model of ^{137}Cs intakes predicted by eqn (3), comparisons were made of predicted body burdens with those that were measured in adults. The time dependence of the radiocesium activity in the body, $Q_k(t)$, normalized by the deposition density, was estimated using

$$Q_k(t) = \int_0^t I_k(\tau) R_k(t-\tau) d\tau \qquad (4)$$

where $R_k(t-\tau)$ = retention function of radiocesium in the body of the k^{th} age group, from ICRP Publication 56 (ICRP 1989).

Fig. 1 shows the predictions of eqn (4) together with normalized WBC measurements of adults from Bragin, Narovlya, and Khoiniki districts in the Gomel Oblast. Results of measurements of adults from the Bryansk Oblast in Russia (Balonov and Travnikova 1993) were also used for comparison. The value of C_0 was selected so that the value of the normalized body burden was 1 Bq per (kBq m^{-2}) for t = 120 d, near the end of August 1986.

Fig. 1 shows that the ^{137}Cs body burdens in adults increased to a maximum during the first 2–3 mo following the accident and then began to decline. Proper estimation of the accumulation of ^{137}Cs during the early period is essential for estimates of internal doses for persons who were evacuated from the 30-km zone to uncontaminated areas during the first ten days after the accident. Doses to evacuees from radiocesiums were calculated using eqn (1), taking into account the variation of intakes given by eqn (3). This approach was used

Table 1. Parameters of modified Garner model.[a]

Parameter index, j	a_j	λ_j (d^{-1})
1	2.9	1.84
2	−5.6	0.69
3	−12.5	0.17
4	13.5	0.05
5	1.5	0.023
6	1.34	0.0072

[a] From Savkin et al. (1996).

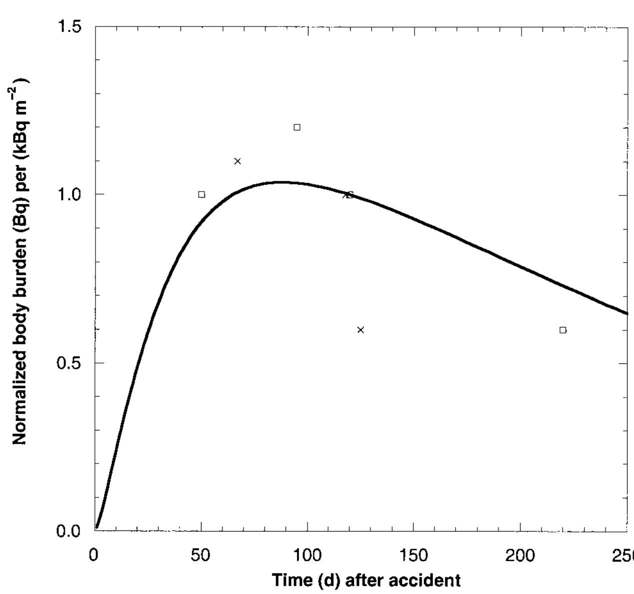

Fig. 1. Time dependence of normalized ^{137}Cs body burdens for adult residents of contaminated areas. Measurements in Gomel Oblast, Belarus (crosses) and Bryansk Oblast, Russia (open squares), and values calculated using eqn (4) (solid line).

because there were no WBC measurements of the population in Belarus prior to 26 June 1986.

Whole-body counting measurements of residents from 12 locations in the Bragin, Narovlya, and Khoiniki districts of Gomel Oblast during July–August 1986 were used to estimate the product [$K\ \nu_k\ C_0$]. These residents reported that they did not markedly change their lifestyles after the accident. The following equation, derived from eqn (3), was used:

$$Kv_kC_0 = \frac{\sigma^{0.5}_{137,nv}Q(t_{meas})}{\int_0^{t_{meas}} \sum_{j=1}^{6} a_j e^{-\lambda_j \tau} R_k(t-\tau)d\tau} \qquad (5)$$

where t_{meas} = time of measurement of ^{137}Cs body burdens; and $Q(t_{meas})$ = median normalized body burden of adult residents, Bq per (kBq m^{-2}).

For all these locations, the derived values of the product [$K\ \nu_k\ C_0$] were found to lie between 0.076 and 0.28 (Bq d^{-1}) per (kBq m^{-2}). The median value of [$K\ \nu_k\ C_0$] of 0.16 (Bq d^{-1}) per (kBq m^{-2}) was used in subsequent calculations. According to the earlier study by Savkin et al. (1996), the average daily consumption of milk by adults in Gomel Oblast was 0.5 L, and it accounted for about 90% of the adult radiocesium intake during this period. Thus, K = 1.1, and C_0 can be estimated to be 0.3 (Bq L^{-1}) per (kBq m^{-2}).

Table 2 contains the parameters that were used for dose estimation for two age groups of children. It was assumed that nearly all the radiocesium intake was due to milk consumption. The initial activity ratio of ^{134}Cs to ^{137}Cs of 0.52 was used to estimate the initial normalized intakes of ^{134}Cs, which were reduced by decay more rapidly than those of ^{137}Cs.

Estimation of the radiocesium intake from late 1986 to January 1992

At later times, doses due to intakes of ^{134}Cs declined, and the contribution of ^{137}Cs was an increasing fraction of the total. The intakes of ^{137}Cs and the body burdens of inhabitants of contaminated territories depend on agricultural and radioecological conditions of the region, on the migration and sorption processes of radionuclides in soil, and on the transfer of radionuclides to foodstuffs. For a significant fraction of the population of Belarus, consumption of locally produced milk was the most important pathway for radiocesium ingestion (Minenko et al. 1996). The radiocesium transport in a region can be characterized by the soil-to-milk transfer factor, defined as the quotient of the concentration of ^{137}Cs in locally produced milk (Bq L^{-1}) to the deposition density of that radionuclide in soil (kBq m^{-2}). The contaminated territory of Belarus was subdivided according to the reference value of the transfer factor, determined in 1991–1992. Transfer of cesium from soil to milk was categorized as low, intermediate, or high for areas in Belarus characterized by reference transfer factors that were, respectively, $<$0.3, 0.3–1.0, or $>$1.0 (Bq L^{-1}) per (kBq m^{-2}). The mean values for these ranges were 0.13, 0.54, and 2.35 (Bq L^{-1}) per (kBq m^{-2}), respectively. These categories are not generic; in Ukraine, there is a much broader range of reference transfer factors (Likhtarev et al. 2000).

Ingestion of ^{137}Cs by the inhabitants also depends on the origin of consumed foodstuffs (Drozdovitch and Minenko 1996). Private farms were the main sources of food for those living in rural settlements and for a fraction of the residents in towns, while the trade network was characteristic for the majority of residents of towns and for residents of cities.

Intake functions for ^{137}Cs were derived for young children (0–6 y) and older children (7–17 y) according to the following geographical and radioecological characteristics:

- Place of residence: rural settlement, town, or city; and
- Magnitude of soil-to-milk transfer factor for location.

Intakes of ^{137}Cs by children in rural areas. Children and other persons residing in rural settlements consumed foodstuffs obtained primarily from small private farms. Their body burdens are therefore related to the ^{137}Cs concentrations in foodstuffs produced locally and, in particular, to the ^{137}Cs concentrations in local milk.

The measured concentrations of ^{137}Cs in milk produced on private farms during the period from August 1986 to October 1995, normalized to the local ^{137}Cs deposition densities, are shown in Figs. 2–4 for the three categories of soil-to-milk transfer factors. The values plotted are the arithmetic mean milk concentrations measured by the Sanitary Epidemiologic Service (SES) during summer months divided by the local ^{137}Cs deposition density. The vertical bars show the range between the 16th and 84th percentiles of the distributions of normalized milk concentrations for each time.

Table 2. Parameters for calculations of internal doses from radiocesium in children.

Parameter	Age group	
	0–6 y	7–17 y
$DF_{Cs\text{-}137}$, Gy Bq^{-1}	0.9×10^{-8}	1.1×10^{-8}
$DF_{Cs\text{-}134}$, Gy Bq^{-1}	1.3×10^{-8}	1.6×10^{-8}
ν_k, L d^{-1}	0.7	0.7
K, dimensionless	1	
C_0, (Bq L^{-1}) per (kBq m^{-2})	0.3	

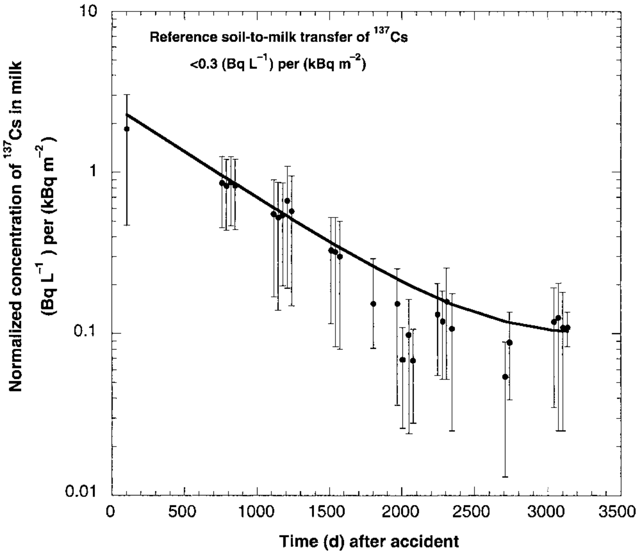

Fig. 2. Normalized concentration of ^{137}Cs in milk produced on private farms in territories with low [<0.3 (Bq L^{-1}) per (kBq m^{-2})] soil-to-milk transfer factors. Ranges shown are the 16th and 84th percentiles of the distributions.

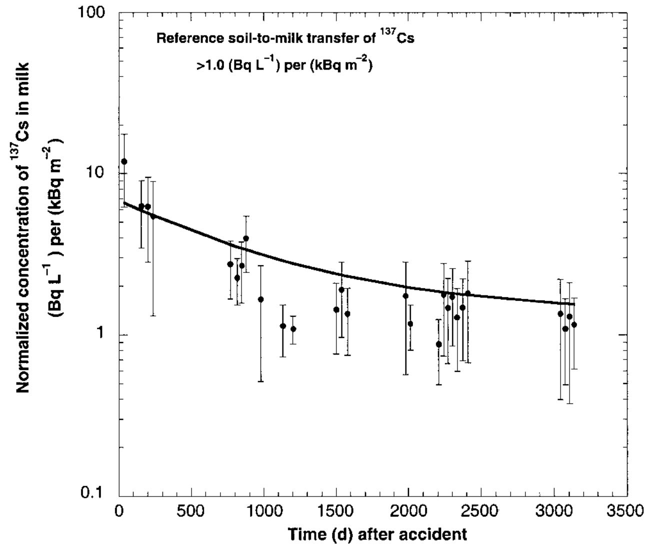

Fig. 4. Normalized concentration of ^{137}Cs in milk produced on private farms in territories with high [>1.0 (Bq L^{-1}) per (kBq m^{-2})] soil-to-milk transfer factors. Ranges shown are the 16th and 84th percentiles of the distributions.

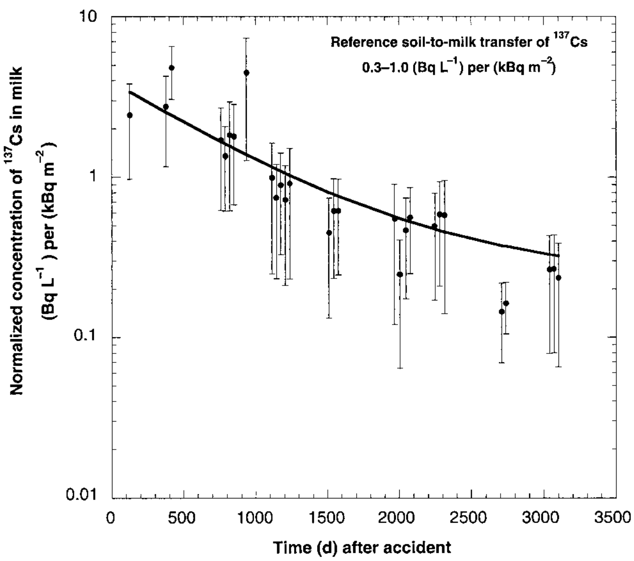

Fig. 3. Normalized concentration of ^{137}Cs in milk produced on private farms in territories with intermediate [0.3–1.0 (Bq L^{-1}) per (kBq m^{-2})] soil-to-milk transfer factors. Ranges shown are the 16th and 84th percentiles of the distributions.

These experimental data can be satisfactorily described by the sum of two exponential functions representing the short- and long-term changes in the normalized milk concentration:

$$C_m(t) = a_{m1}e^{-(\lambda_{m1}+\lambda_r)} + a_{m2}e^{-(\lambda_{m2}+\lambda_r)} \quad (6)$$

where

$C_m(t)$ = a function that describes the time dependence of the normalized concentration of ^{137}Cs in milk, (Bq L^{-1}) per (kBq m^{-2});

a_{m1}, a_{m2} = best fit parameters (Bq L^{-1}) per (kBq m^{-2}) derived by fitting the function to the data using the technique of least square deviations;

λ_{m1}, λ_{m2} = best fit rate constants (d^{-1}) derived by fitting the function to the data using the technique of least square deviations; and

λ_r = radioactive decay rate constant (d^{-1}) for ^{137}Cs.

The best-fit parameter values obtained for the territories with different soil-to-milk transfer factors are presented in Table 3. The parameters λ_{m1} and λ_{m2} are the rate constants that describe the short- and long-term changes in the concentration of ^{137}Cs in milk. As can be seen from Table 3, in spite of the fact that territories with different radioecological conditions were considered, there is no substantial difference between the values of the short-term component of the rate of elimination of ^{137}Cs from milk, λ_{m1}. For all territories, the half-times for the short-term component of the normalized concentration of ^{137}Cs in milk were 1.4–1.6 y. A comparable analysis by Fesenko et al. (1995) of the time dependence of ^{137}Cs concentrations in milk in the Bryansk Oblast of Russia, adjacent to Belarus, yielded estimates of 0.8–1.3 y for the ^{137}Cs removal half-time.

It was observed that the best-fit rate constant λ_{m2} for territories with low soil-to-milk transfer indicates a

Table 3. Best-fit constants for functions describing the time dependence of the normalized concentrations of ^{137}Cs in milk in rural areas.

Soil-to-milk transfer factor (Bq L^{-1}) per (kBq m^{-2})	N_{meas}[a]	N_{set}[b]	Best-fit constants: (Bq L^{-1}) per (kBq m^{-2}) a_{1m}	a_{2m}	λ_{1m} (d^{-1})	λ_{2m} (d^{-1})
<0.3	6,264	745	2.6	0.01	1.34×10^{-3}	-5.9×10^{-4}
0.3–1.0	3,192	260	3.6	0.37	1.32×10^{-3}	1.1×10^{-4}
>1.0	3,946	237	5.1	1.7	1.22×10^{-3}	5.4×10^{-5}

[a] Number of measurements.
[b] Number of settlements.

long-term increase in the normalized concentration of ^{137}Cs in milk. The corresponding half-time is 3.2 y. The contribution of the long-term component to the ^{137}Cs concentration in milk is very small. Estimation of this half-time is much more uncertain because the period of observation of the long-term component is rather short. Unmeasured uncertainties related to non-uniform ^{137}Cs deposition densities on pastures, sampling variability, and incomplete sample design might result in this unexpected tendency for a long-term increase. Eqn (6) should not be extrapolated to times after October 1995, the last period of observation.

^{137}Cs body burdens in rural children. Under the assumption that a single exponential retention function is adequate to describe the ^{137}Cs body burden in children, the following differential equation applies:

$$\frac{dq}{dt} = I(t) - (\lambda_b + \lambda_r)q \qquad (7)$$

where q = ^{137}Cs activity in the body (Bq); $I(t)$ = rate of intake of ^{137}Cs (Bq d^{-1}); and λ_b = rate constant (d^{-1}) corresponding to the biological half-time (T_b, d) of ^{137}Cs in the body.

Estimates of the biological half-times of ^{137}Cs in children of age α (y) were obtained using the equation of Baverstock (1987):

$$T_b(\alpha) = 11.9 + 1.27M(\alpha) \qquad (8)$$

in which $M(\alpha)$ is the average whole-body mass (kg) of children of that age. Average values of T_b of 64 d for children aged 7–17 y and 30 d for younger children (0–6 y) were calculated using eqn (8) and whole-body masses given in ICRP Publication 56 (ICRP 1989). The corresponding values of λ_b are 0.0108 and 0.0231 d^{-1}, respectively. The computed half-times are about 20% greater than average half-times for those age groups estimated from serial whole-body counts of children from the Bryansk Oblast in Russia by Lebedev and Yakovlev (1993). Averages of their values for ages (0–6 y) and (7–17 y) are 24 and 55 d, respectively.

Because milk and milk products were the main contributors to ^{137}Cs ingestion by rural residents, it is reasonable to assume that the ^{137}Cs intake function $I(t)$ is similar in form to eqn (6), which describes the ^{137}Cs concentration in milk. Eqn (7) can be solved for an intake function of that form for the initial condition that $q(0)$ = 0. Expressed in terms of the normalized body burden, $Q(t)$, with units Bq per (kBq m^{-2}), the solution is:

$$Q(t) = a_1 e^{-(\lambda_1+\lambda_r)t} + a_2 e^{-(\lambda_2+\lambda_r)t} - a_3 e^{-(\lambda_b+\lambda_r)t} \qquad (9)$$

Whole-body counting measurements of children living in rural areas were performed by the Institute of Oncology and Medical Radiology in Minsk, by the Institute of Radiation Medicine and Endocrinology in Minsk, and by local district hospitals. These WBC data have been compiled in a database by the Institute of Radiation Medicine and Endocrinology. The whole-body counting results have been normalized to the local ^{137}Cs deposition densities for the residence locations of the children who were counted. More than 40,000 normalized WBC measurements have been used to derive parameters a_1, a_2, and a_3 [Bq per (kBq m^{-2})] and λ_1, λ_2 (d^{-1}) for two age groups of children living in areas having low, intermediate, or high soil-to-milk transfer of ^{137}Cs.

Values of the best-fit parameters, derived from a least squares procedure, are given in Table 4 together with the numbers of measurements used to derive each set of parameters. For the regions of low soil-to-milk transfer, the negative values of λ_2 suggest long-term increases in the normalized body burdens of both age groups. This may reflect a greater reliance on natural foods in later years, but the caveats mentioned earlier for the normalized milk concentration in such areas apply here as well.

Comparisons of normalized body burdens (Bq per kBq m^{-2}) estimated using the parameters given in Table

Table 4. Parameters of functions describing the time dependence of normalized ^{137}Cs body burdens in children living in rural areas.

Parameter	Age group 0–6 y	7–17 y
Reference soil-to-milk transfer of ^{137}Cs <0.3 (Bq L^{-1}) per (kBq m^{-2})		
a_1, Bq per kBq m^{-2}	28.9	247.5
a_2, Bq per kBq m^{-2}	3.2	9.3
a_3, Bq per kBq m^{-2}	32.1	256.8
λ_1, d^{-1}	1.86×10^{-3}	3.06×10^{-3}
λ_2, d^{-1}	-2.1×10^{-4}	-1.0×10^{-5}
Number of measurements	2,543	14,226
Reference soil-to-milk transfer of ^{137}Cs of 0.3–1.0 (Bq L^{-1}) per (kBq m^{-2})		
a_1, Bq per kBq m^{-2}	57.4	238.7
a_2, Bq per kBq m^{-2}	8.7	22.5
a_3, Bq per kBq m^{-2}	66.1	261.2
λ_1, d^{-1}	2.12×10^{-3}	2.36×10^{-3}
λ_2, d^{-1}	5.0×10^{-5}	9.0×10^{-5}
Number of measurements	2,967	15,906
Reference soil-to-milk transfer of ^{137}Cs of 1.0–5.0 (Bq L^{-1}) per (kBq m^{-2})		
a_1, Bq per kBq m^{-2}	191	772.7
a_2, Bq per kBq m^{-2}	17	31.2
a_3, Bq per kBq m^{-2}	208	803.9
λ_1, d^{-1}	2.0×10^{-3}	1.95×10^{-3}
λ_2, d^{-1}	6×10^{-5}	1.2×10^{-4}
Number of measurements	1,123	6,101

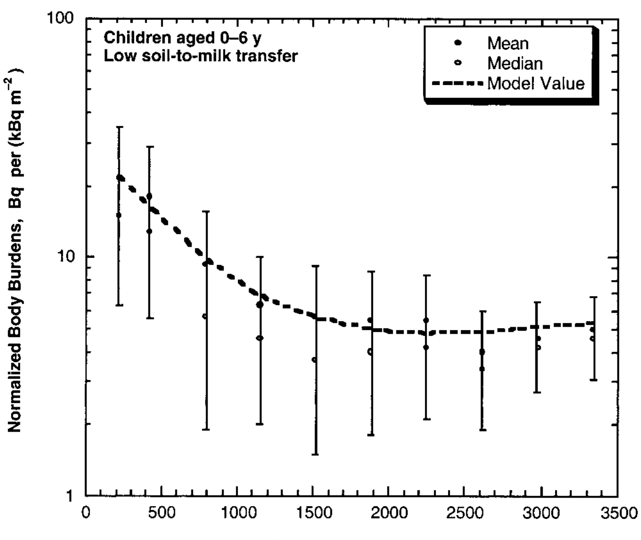

Fig. 5. Normalized ^{137}Cs body burdens for children aged 0–6 y living in areas with low soil-to-milk transfer factors. Ranges shown are the 16th and 84th percentiles of the distributions. The line indicates estimates obtained using eqn (9) and parameters from Table 4.

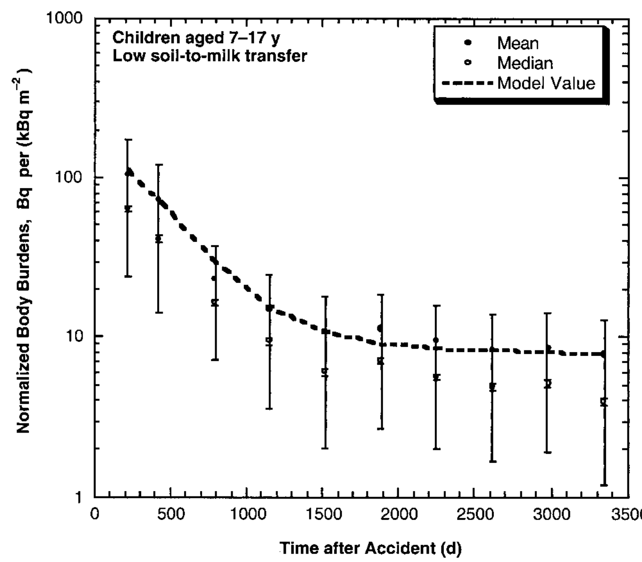

Fig. 6. Normalized ^{137}Cs body burdens for children aged 7–17 y living in areas with low soil-to-milk transfer factors. Ranges shown are the 16th and 84th percentiles of the distributions. The line indicates estimates obtained using eqn (9) and parameters from Table 4.

4 and measurements of ^{137}Cs activity in the two groups of children are shown in Figs. 5–8. The comparisons begin at 220 d after the accident, late in 1986, when substantial numbers of measurements of body burdens are available for children in rural areas. The first two figures are for areas of low soil-to-milk transfer of ^{137}Cs, while Figs. 7 and 8 are for areas of intermediate soil-to-milk transfer. In the figures, the median and mean normalized body burdens are shown together with the 16th and 84th percentiles of the distributions. More than 50 children

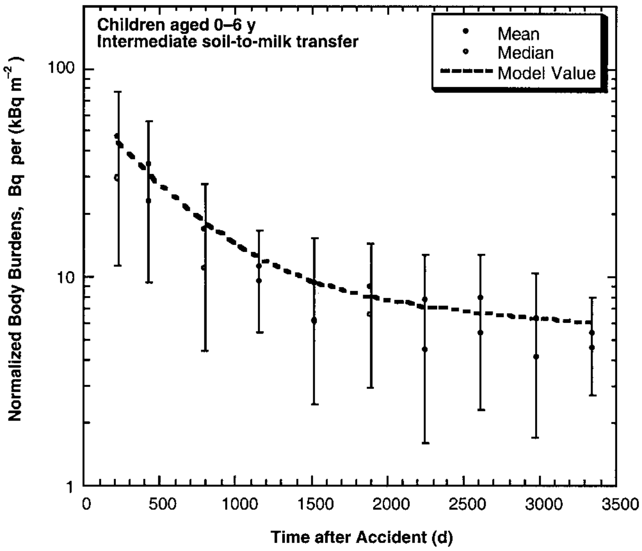

Fig. 7. Normalized ^{137}Cs body burdens for children aged 0–6 y living in areas with intermediate soil-to-milk transfer factors. Ranges shown are the 16th and 84th percentiles of the distributions. The line indicates estimates obtained using eqn (9) and parameters from Table 4.

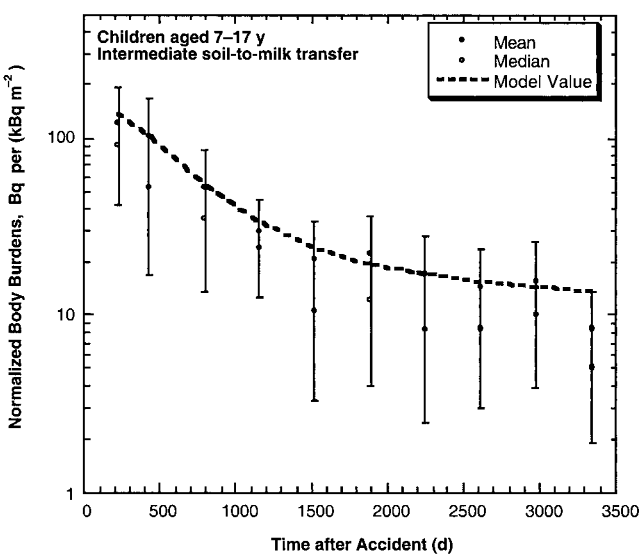

Fig. 8. Normalized ^{137}Cs body burdens for children aged 7–17 y living in areas with intermediate soil-to-milk transfer factors. Ranges shown are the 16th and 84th percentiles of the distributions. The line indicates estimates obtained using eqn (9) and parameters from Table 4.

aged 0–6 y were measured in all years except in 1988 for low soil-to-milk transfer areas and in 1995 for intermediate soil-to-milk transfer areas. More than 300 children aged 7–17 y were measured in all years except in 1988 for low soil-to-milk transfer areas and in 1986 and 1995 for intermediate soil-to-milk transfer areas.

^{137}Cs body burdens in urban children. Sources of foodstuffs consumed by children in the urban areas are different. Two types of urban settlements can be distinguished: large cities, where inhabitants mainly purchased foodstuffs in state shops, and towns where inhabitants consumed foodstuffs from the trade network as well as from their private farms. Milk is again the most important source of ^{137}Cs in the diet.

The results of measurements of ^{137}Cs concentration in milk performed in 1990–1994 are given in Table 5. Samples were taken from the trade network in the cities of Gomel and Mogilev and in towns. Measurements were also made for private farms in towns and rural settlements. The towns and rural settlements from which samples were collected are located in the contaminated territories of the neighboring districts of the northern part of Gomel Oblast and of the southern part of Mogilev Oblast. The reference soil-to-milk transfer factor for ^{137}Cs in these areas is low, <0.3 (Bq L^{-1}) per (kBq m^{-2}).

The data from Table 5 indicate that levels of ^{137}Cs concentration in milk from the trade network in cities and towns are very similar and are not correlated with the local ^{137}Cs deposition density. There was more variability in the concentrations of ^{137}Cs in milk from private farms, as is shown by the relatively large standard deviations. Some of the concentration distributions appear to be lognormal. The ^{137}Cs concentrations in locally produced milk from private farms in and near towns are similar to ones in rural settlements. The soil contamination levels for these areas were comparable. On the basis of the data in Table 5, it is assumed that the inhabitants of towns who consumed foodstuffs from their private farms can be considered to be part of the rural population.

A function of the type given by eqn (9) was fit to data on normalized body burdens of nearly 40,000 urban children. Approximately 80% of the measurements were for the older age group; even so, there were more than 7,800 children in the younger group. The best-fit parameters are given in Table 6. The estimated half-times for the short-term component of normalized uptake of ^{137}Cs by inhabitants of towns of 0.8–1.3 y were obtained. This range is close to that found for rural populations. The suggestion of a long-term increase in normalized body burdens is seen in this group as well.

In general, similar changes in ^{137}Cs body burdens were observed for adult inhabitants of cities, towns, and rural settlements during the period 1986–1992. Average values of body burdens measured in adults are given in Table 7. In cities, practically no foodstuffs are produced locally; thus, as was shown above, the ^{137}Cs concentrations in milk and other foodstuffs from the trade network are not highly correlated with the local ^{137}Cs deposition

Table 5. ^{137}Cs concentrations in milk from the trade network and private farms.

	^{137}Cs concentrations in milk (Bq L^{-1})[a]			
	Trade network		Private farms	
	Towns	Cities	Rural	Towns
Soil contamination	450 kBq m^{-2}	50 kBq m^{-2}	410 kBq m^{-2}	450 kBq m^{-2}
1990		22 ± 16 (18)		
1991	11 ± 2 (4)	18 ± 7 (22)	76 ± 84 (745)	98 ± 78 (44)
1992		11 ± 2 (51)	38 ± 47 (26)	
1994	13 ± 10 (9)	11 ± 8 (2,708)	58 ± 48 (3,286)	80 ± 77 (50)

[a] Mean ± standard deviation, with number of measurements shown in parentheses.

Table 6. Parameters of function describing the time dependence of normalized ^{137}Cs body burden for children living in towns (1986–1995).

	Age group	
Parameter	0–6 y	7–17 y
a_1, Bq per kBq m^{-2}	20	65
a_2, Bq per kBq m^{-2}	3	5
a_3, Bq per kBq m^{-2}	23	70
λ_1, d^{-1}	1.4×10^{-3}	2.3×10^{-3}
λ_2, d^{-1}	-4.6×10^{-4}	-3.0×10^{-4}
Number of measurements	7,850	32,112

densities. Moreover, foodstuffs sold in state shops are monitored by the radiological control system.

When contamination limits were exceeded, foods were removed from the shops. The levels of control for milk, which changed with time, are shown in Table 8. The operation of control procedures in cities means that normalization of ^{137}Cs body burden to ^{137}Cs soil deposition density cannot be applied for that population group. To estimate internal exposure from radiocesium ingestion to inhabitants of cities, the results of cesium body burden measurements were used directly. For rural settlements and towns, normalized milk concentrations were used as the basis for estimation of body burdens and internal exposures from ^{137}Cs and ^{134}Cs.

METHODS FOR ESTIMATING DOSES DUE TO EXTERNAL IRRADIATION

Portions of the total thyroid doses received by the study subjects were due to exposure to external gamma radiation from radionuclides deposited on the ground. The methods used for estimating those doses are presented in this section. In general, assessment of absorbed doses from external radiation sources in the environment requires knowledge of several factors that influence the dose estimates. These factors include:

- isotopic composition of the radioactive fallout that was deposited;
- whether the deposition was primarily due to wet or dry processes;
- the soil density and the rates of radionuclide migration into the soil;
- information about the subject's residence location and lifestyle; and
- information about the type of dwelling and the settlement in which it was located.

The principal source of external exposure was from gamma rays emitted by radionuclides deposited on the ground. External exposure from airborne radionuclides during the period of cloud passage was neglected. It was also assumed that noble gases in the passing cloud were not deposited on ground surfaces. The period of cloud passage was brief in comparison to the period of external exposure, which was from the date of deposition until 31 December 1991.

Because the period of integration of the external doses was long (~5.7 y), the calculations account for the effects of downward migration of the radionuclides in the soil. The depth distribution of radionuclide activity in the soil has been assumed to be exponential in form:

$$A_m(z,t) = A_m(0,0)e^{-\frac{z}{\beta(t)}} \qquad (10)$$

where

z = depth in soil (g cm^{-2});
$A_m(z, t)$ = activity concentration at depth z at time t (y); and
$\beta(t)$ = parameter (g cm^{-2}) describing the exponential distribution of activity with depth.

The distribution parameter is considered to be time dependent, increasing over time from a small initial value:

$$\beta(t) = \beta(0) + \upsilon t \qquad (11)$$

where

t = time (y) after initial deposition of contamination;

Table 7. Measured ^{137}Cs body burdens for adults in the Belarusian population.

Year	^{137}Cs body-burden (kBq)[a,b]		
	Cities[c]	Towns	Rural settlements
1986	no estimate	28 ± 20 (367)	129 ± 148 (3,328)
1987	10 ± 3.6 (16)	17 ± 14 (117)	59 ± 107 (4,498)
1988	4.1 ± 6.2 (20)	6.4 ± 9.6 (606)	25 ± 46 (3,266)
1989	3.4 ± 4.9 (1,405)	6.3 ± 6.2 (5,283)	13 ± 20 (10,652)
1990	1.8 ± 1.6 (3,551)	4.6 ± 6.6 (5,989)	7.2 ± 11.1 (8,232)
1991	1.0 ± 2.1 (11,458)	4.8 ± 9.4 (2,994)	6.4 ± 9.5 (4,617)
1992	1.0 ± 2.0 (7,155)	3.5 ± 7.0 (5,264)	6.4 ± 17.9 (5,921)
1993	0.8 ± 1.4 (1,266)	2.8 ± 4.9 (4,679)	4.4 ± 7.0 (5,414)
1994	1.1 ± 2.6 (2,001)	6.7 ± 12.8 (5,488)	7.1 ± 11.8 (5,098)
1995	1.0 ± 1.4 (530)	6.3 ± 7.3 (361)	6.9 ± 10.4 (392)

[a] Values of the mean ± the standard deviation of all results for the year.
[b] Number of measurements shown in parentheses.
[c] Cities (Gomel, Mogilev, and Mozyr) are located in areas with ^{137}Cs soil contamination >37 kBq m^{-2}.

Table 8. Control levels for concentrations of ^{137}Cs in milk.

Period	Control level (Bq L^{-1}) for ^{137}Cs concentration in milk
6–30 May 1986	3,700
After 30 May 1986	370

$\beta(0)$ = value (g cm^{-2}) of distribution parameter immediately after fallout; and
υ = average rate of downward migration (g cm^{-2} y^{-1}).

Values of parameters in eqn (11) were determined using the results of environmental measurements and soil samples at undisturbed contaminated sites. Korneev et al. (1996) found that an average rate of downward migration of 0.18 g cm^{-2} y^{-1} was appropriate for cesium deposited on the territory of Belarus. They also estimated values of β_0 of 0.5 g cm^{-2} for wet deposition and 0.2 g cm^{-2} for dry deposition in Belarus.

Typically, the external exposure rates at outdoor locations in a settlement are lower than those over open undisturbed terrain. This fact is accounted for through an empirical dose rate reduction factor, *RF* (dimensionless). A value of $RF = 0.7$ was obtained (Minenko et al. 1996) by averaging over data for settlements located in radioactively contaminated areas in Belarus and that result is used here. External radiation doses to the members of public are further reduced due to shielding provided by dwellings and other buildings. The dimensionless shielding factor, *SF*, accounts for the reduction of dose due to time spent indoors in a residence and seasonal variations of exposure rate. Table 9 contains estimates made by Minenko et al. (1996) that are based upon results of TLD measurements. Values of the shielding factor for the two age groups in the rural population differed in 1986. There was a smaller difference for children in towns during that year, but no difference in cities. In later years (1987–1995), the differences between the age groups are small. There are differences among residence locations for both 1986 and the later period.

Table 9. Values of the shielding factor for different ages, residence locations, and times.

Age group	Values of the shielding factor (*SF*)					
	Rural areas		Towns		Cities	
	1986	1987–1995	1986	1987–1995	1986	1987–1995
0–6 y	0.42	0.36	0.33	0.28	0.30	0.24
7–17 y	0.48	0.37	0.36	0.30	0.30	0.24

In computations of the thyroid exposure to external dose the combined reduction factor (*RF**), the product of *RF* and *SF*, is used. For the present calculations, a generally conservative value $RF^* = 0.33$ was selected. It is intermediate between the two values for rural settlements in 1986 and only slightly lower than the combined reduction factor for older children during that year.

The mixture of radionuclides in the deposited radioactivity has an important influence on the external dose. The radionuclide composition of Chernobyl fallout in Belarus varied considerably (Izrael et al. 1990; Orlov et al. 1992; Buzulukov and Dobrynin 1993; Petriaev 1994). Unfortunately, experimentally measured isotopic ratios data are only available for a limited number of settlements. The present paper follows the approach in the companion paper (Gavrilin et al. 2004), in which Belarus was divided into ten regions according to the type of deposition (wet or dry), available information about contamination levels, and radionuclide ratios. The radionuclide ratios adopted for use in external thyroid dose calculations are presented in Table 10. Such a scheme inevitably leads to the appearance of stepwise changes at the boundaries of regions, which do not occur in reality.

Table 10. Radionuclide activities relative to ^{137}Cs activity used in calculations of thyroid doses from external exposure.

	Radionuclide activity relative to ^{137}Cs activity at the time of the main deposition in the region									
Radionuclide	1 30-km zone[b]	2 Central spot-N[c]	3[a] NE spot[d]	4 Gomel city	5[a] G-M Rem.[e]	6[a] Mogilev city	7 Central spot-W[f]	8[a] Minsk city	9 BR MR, G[g]	10 Vit. Obl.[h]
^{95}Zr	2.4	3.6	0.17	1.3	4.0	0.34	0.77	0.3	0.5	0.2
^{95}Nb	*eq*[i]	*eq*	*eq*	*eq*	*eq*	*eq*	*eq*	*eq*	*eq*	*eq*
^{99}Mo	9.3	7.7	2.0	4.4	4.4	2.4	2.8	2.4	2.8	1.4
^{103}Ru	3.3	3.7	1.6	2.8	2.4	1.9	2.2	1.5	2.8	1.8
^{106}Ru	0.71	0.93	0.42	1.0	0.85	0.34	0.86	0.45	0.7	0.4
^{131m}Te	1.3	0.64	0.83	0.14	0.23	0.23	0.53	0.11	0.78	0.01
^{132}Te	10	8.0	11	2.6	4.2	4.2	7.6	2.8	12	4.8
^{131}I	13	16	8.3	17	21	21	19	14	23	24
^{132}I	*eq*	*eq*	*eq*	*eq*	*eq*	*eq*	*eq*	*eq*	*eq*	*eq*
^{133}I	14	7.5	3.4	3.4	5.0	5.0	6.8	2.0	8.3	1.2
^{135}I	5.2	1.0	0.12	0.017	0.13	0.13	0.23	0.014	0.55	0.19
^{134}Cs	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
^{136}Cs	0.3	0.27	0.27	0.26	0.26	0.26	0.27	0.3	0.3	0.3
^{137}Cs	1	1	1	1	1	1	1	1	1	1
^{140}Ba	5.0	4.7	0.76	6.0	7.6	1.3	2.1	1.0	1.5	0.8
^{140}La	*eq*	*eq*	*eq*	*eq*	*eq*	*eq*	*eq*	*eq*	*eq*	*eq*
^{141}Ce	3.1	3.9	0.14	2.4	3.8	0.33	0.63	0.25	0.5	0.2
^{144}Ce	2.0	2.8	0.12	1.1	3.0	0.27	0.49	0.18	0.3	0.15
^{239}Np	3.0	4.0	0.2	2.4	4.0	0.3	0.7	0.3	0.5	0.2

[a] Deposition in this zone was primarily by wet processes.
[b] The portion of the 30-km zone that lies in Belarus.
[c] The northern part of the central spot of contamination includes Bragin, Khoiniki, Yelsk, Mozyr, and Narovlya districts and parts of Kalinkovichi, Lelchitsa, Loev, and Rechitsa districts in the Gomel Oblast.
[d] The northeastern spot, or Gomel-Mogilev spot, includes Buda-Koshelev, Chechersk, Dobrush, Korma, and Vetka districts in Gomel Oblast and Bikhov, Cherikov, Krasnopolye, and Slavgorod districts in Mogilev Oblast.
[e] Remaining portions of the Gomel and Mogilev Oblasts not included in regions 1, 2, 3, 4, and 6.
[f] The western part of the central spot includes Luninets, Pinsk, and Stolin districts in Brest Oblast.
[g] The remaining portions (not in regions 7 and 9) of the Brest and Minsk Oblasts and Grodno Oblast.
[h] Vitebsk Oblast.
[i] *eq* denotes that the radionuclide activity was assumed to be in transient or secular equilibrium with the precursor radionuclide.

Because the deposition density of the long-lived radionuclide ^{137}Cs has been measured in many locations in Belarus since 1986, this approach allows estimation of the external doses for all settlements where subjects resided.

The thyroid dose due to environmental contamination by the radionuclides listed can be expressed in the following way:

$$D_{p,\xi} = RF^{*}\sigma_{137,\xi}\int_{0}^{T_{cc}} DF_{p,i}(t)A_i(t)dt \qquad (12)$$

where

$D_{p,\xi}$ = thyroid dose to the person p living in settlement ξ;
$\sigma_{137,\xi}$ = deposition density of ^{137}Cs in settlement ξ;
$A_i(t)$ = ratio of the activity of the i^{th} radionuclide to the ^{137}Cs activity in soil at time t;
$DF_{p,i}$ = absorbed dose rate in thyroid per unit activity of the i^{th} radionuclide in soil, Gy y^{-1} per Bq m^{-2}; and
T_{cc} = 5.7 y, the time between the date of deposition and the case-control study reference date (31 December 1991).

In most calculations employing eqn (12), it was assumed that the person lived in the same settlement throughout the period of integration. To account for travel away from the settlement, including permanent relocations or temporary evacuations, it is necessary to have detailed individual information for each subject. Such detailed individual information is not available for all subjects; however, it is known that inhabitants of the 30-km zone (region 1) were evacuated. In some cases, the time of evacuation and the destination are known. The evacuation time was not known for one subject: for the calculations it was assumed to be 7 d after the accident, an average time according to responses to subsequent questioning of evacuees. The destination was not known for two subjects who were evacuated; for the calculations it was assumed to be to a place with a mean ^{137}Cs deposition density of ~185 kBq m^{-2}, typical of reported relocation destinations.

During the 5.7-y period, other subjects may have moved temporarily from their original residences in other regions to areas of lower or higher contamination. Permanent moves are considered less likely than temporary moves corresponding to school vacations but may also have occurred. Such changes in residence are not reflected in the dose estimates presented here.

The thyroid dose conversion factor $DF_{p,i}(t)$ depends upon a person's age (α), and place of residence in one of the regions (ζ) shown in Table 10. A different function, $DF_i\ (t,\alpha,\zeta)$, may be defined for each radionuclide. To estimate appropriate values of the dose conversion factors for the post-accident situation, a procedure described by Eckerman and Ryman (1993) has been used. Following that approach and assuming the shape of the activity distribution given in eqn (10), the dose conversion factors for individual radionuclides can be represented as:

$$DF_i(t,\alpha,\zeta) = \sum_{j=1}^{N} \frac{Y_{i,j}}{\beta(t)} AC(\alpha,E_{i,j}) \int_0^\infty PF(E_{i,j},z) e^{-\frac{z}{\beta(t)}} dz \tag{13}$$

where

$Y_{i,j}$ = quantum yield of the j^{th} gamma ray emitted by the i^{th} radionuclide;

$E_{i,j}$ = energy (MeV) of the j^{th} gamma ray emitted by the i^{th} radionuclide;

$\beta(t)$ = time-dependent distribution parameter (g cm^{-2}), eqn (11);

$AC(\alpha,E_{i,j})$ = function that accounts for age dependence of the dose conversion factors for particular age groups (discussed below);

$PF(E_{i,j},z)$ = thyroid dose rate conversion factor for a plane isotropic source of photons of energy $E_{i,j}$ in soil at depth z (Gy y^{-1} per Bq m^{-2}); and

N = the number of gamma rays emitted by the i^{th} radionuclide.

The plane source dose conversion factors, $PF(E_{i,j},z)$, presented in Eckerman and Ryman (1993) are for adults only. Age-dependent dose factors computed by Monte Carlo methods have been provided by Jacob et al. (1990) for a plane source located at a depth z = 0.5 g cm^{-2} to account for soil roughness and initial nuclide migration. This source representation is applicable to a fresh deposition, but it cannot be applied for a time period of several years. Information provided by Jacob et al. (1990) on thyroid dose factors for a 2-mo-old baby and a 7-y-old child has been used to estimate the age dependence of the dose conversion factors. This is not a thoroughly rigorous procedure because of changes in the gamma ray spectrum that occur at different stages of downward migration of radionuclides in the soil. Nevertheless, it appears that the uncertainty introduced by correction factors for age is considerably lower than uncertainties due to modification factors that account for the type of dwelling and personal behavior.

Taking into account ages at the time of the accident of children included in the case-control study, four age groups have been established for calculations of thyroid doses from external exposure. The three main groups are babies, aged 0–1 y, children aged 2–13 y, and older children ($\geq$14 y old). Five persons who were not born yet at the time of the accident comprise the fourth age group. Prior to birth, the appropriate thyroid dose factors were assumed to be equal to the dose factors for the uterus of the mother (Eckerman and Ryman 1993). Based upon the work of Jacob et al. (1990), the following analytical expressions have been derived for use in computations of thyroid dose factors for younger persons:

$$\begin{aligned} AC(\alpha,E > 0.002\text{MeV}) &= \frac{0.031}{E - 0.0094} + 1.15;\ 0 \leq \alpha < 2y \\ AC(\alpha,E > 0.002\text{MeV}) &= \frac{0.0074}{E - 0.0017} + 1.05;\ 2 \leq \alpha < 14y \end{aligned} \tag{14}$$

The subscripts relating E (MeV) to particular gamma rays from individual radionuclides were not included in eqn (14). The thyroid dose factor used in calculations was modified if a subject's age category changed during the 5.7-y period.

Table 11 presents the radionuclides of interest, their half-lives and gamma-ray emission factors. The thyroid dose factors for exposure to an effective plane source located at a depth of 0.5 g cm^{-2} were computed using information from Eckerman and Ryman (1993) and the adjustment factors given in eqn 14. Dose factors for some radionuclides, which have half-lives on the order of hours or less, were considered to be in secular equilibrium with the precursor radionuclide and the contributions from both nuclides were incorporated into a single factor. The decay chains considered have two members, and the time dependent activity ratios can be calculated using eqn 15.

$$A_i(t) = A_i(0)e^{-\lambda_i t} + A_{pr,i}(0) f_i \frac{\lambda_i}{\lambda_i - \lambda_{pr,i}} (e^{-\lambda_{pr,i} t} - e^{-\lambda_i t}) \tag{15}$$

Table 11. Age-dependent thyroid dose factors for radionuclides from effective plane source located at a depth of 0.5 g cm^{-2}.

Radionuclide	Half-life	Principal decay product (yield)	Thyroid dose factor for plane source at depth 0.5 g cm^{-2} (nGy h^{-1}) per (kBq m^{-2})		
			Baby (8 wk)	Child (7 y)	Adult
^{95}Zr	63.98 d	^{95}Nb (0.99)	2.28	2.03	1.91
^{95}Nb	35.15 d		2.36	2.10	1.98
^{99}Mo[a]	66.0 h		0.81	0.69	0.63
^{103}Ru[a]	39.28 d		1.46	1.28	1.21
^{106}Ru[a]	368.2 d		0.61	0.54	0.51
^{131m}Te[a]	30 h	^{131}I (0.78)	4.49	4.00	3.76
^{131}I	8.04 d		1.18	1.03	0.96
^{132}Te	78.2 h	^{132}I (1.0)	0.69	0.57	0.52
^{132}I	2.30 h		6.86	6.11	5.78
^{133}I	20.8 h		1.86	1.65	1.55
^{135}I	6.61 h		4.49	4.03	3.81
^{134}Cs	2.062 y		4.78	4.24	4.00
^{136}Cs	13.1 d		6.54	5.81	5.46
^{137}Cs[a]	30.0 y		1.75	1.55	1.46
^{140}Ba	12.74 d	^{140}La (1.0)	0.56	0.49	0.46
^{140}La	40.272 h		6.62	5.92	5.59
^{141}Ce	32.501 d		0.24	0.19	0.17
^{144}Ce[a]	284.3 d		0.15	0.13	0.12
^{239}Np	2.355 d		0.52	0.42	0.38

[a] Dose factors include contributions from short-lived progeny assuming that they are in radioactive equilibrium.

where

$A_i(t)$ = activity ratio relative to ^{137}Cs of the i^{th} radionuclide at time t;
$A_i(0)$ = initial activity ratio relative to ^{137}Cs of the i^{th} radionuclide;
$A_{pr,i}(0)$ = initial activity ratio relative to ^{137}Cs of the precursor of the i^{th} radionuclide;
f_i = fraction of the decays of the precursor that produce the i^{th} radionuclide;
λ_i = radioactive decay rate constant (y^{-1}) of the i^{th} radionuclide; and
$\lambda_{pr,i}$ = radioactive decay rate constant (y^{-1}) of the precursor of the i^{th} radionuclide.

The activity ratios given by eqn (15) were used in eqn (12) together with the other relevant parameters to estimate the external radiation contribution to thyroid doses received by subjects in the study.

RESULTS

Thyroid doses from ingestion of radiocesium (^{134}Cs and ^{137}Cs) and from external irradiation by deposited radionuclides have been estimated using the methods described above. Results of the calculations are presented and summarized here.

Average thyroid doses from ingestion of radiocesium were calculated for members of two age groups (0–6 y and 7–17 y) located in contamination zones with different soil-to-milk transfer factors. Low and intermediate values of reference soil-to-milk transfer factors [<0.3 and 0.3–1.0 (Bq L^{-1}) per (kBq m^{-2}), respectively] were observed over much of the contaminated territory of Gomel and Mogilev Oblasts. In the territory of Polesye, located in the eastern part of Brest Oblast and the southwestern part of Gomel Oblast, reference soil-to-milk transfer factors >1.0 (Bq L^{-1}) per (kBq m^{-2}) were found. Cumulative thyroid doses from radiocesium ingestion, normalized to ^{137}Cs deposition density, are plotted in Figs. 9 and 10 for towns and for rural areas

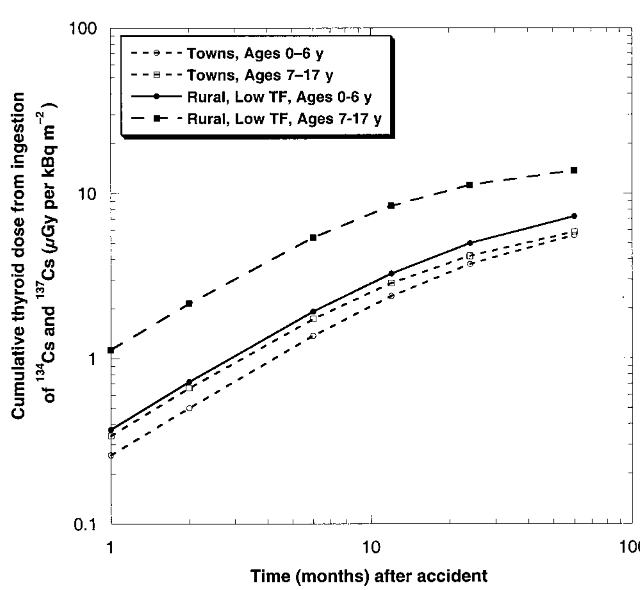

Fig. 9. Normalized cumulative thyroid doses from ingestion of radiocesium by children in towns and in rural areas with low soil-to-milk transfer factors [<0.3 (Bq L^{-1}) per (kBq m^{-2})].

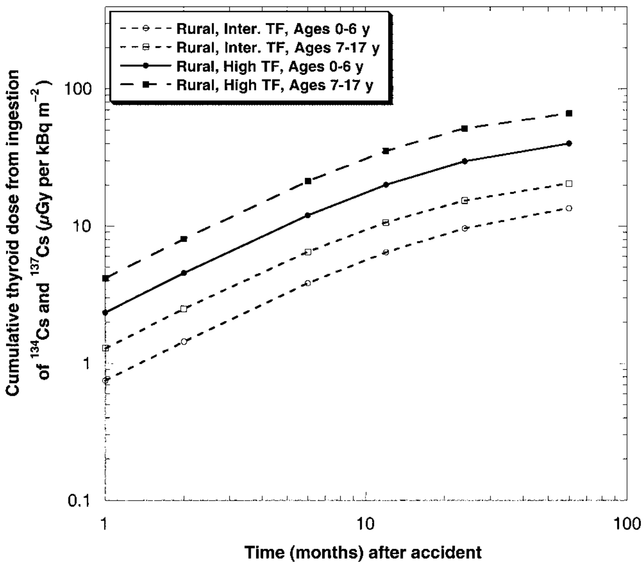

Fig. 10. Normalized cumulative thyroid doses from ingestion of radiocesium by children in rural areas with intermediate [0.3–1.0 (Bq L^{-1}) per (kBq m^{-2})] and high [>1.0 (Bq L^{-1}) per (kBq m^{-2})] soil-to-milk transfer factors.

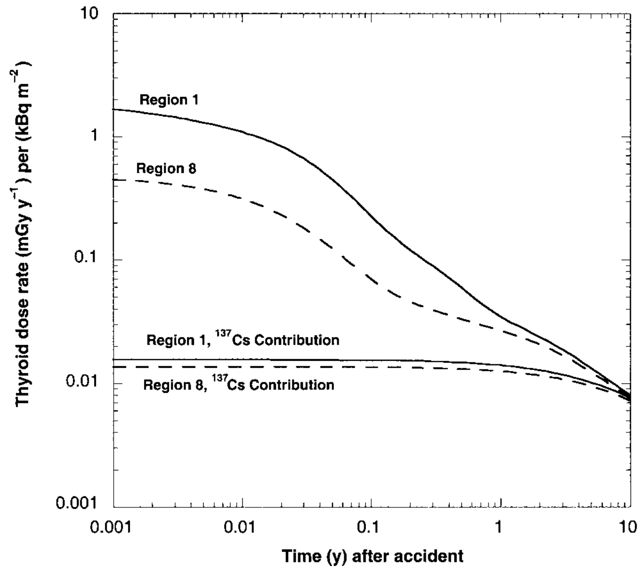

Fig. 11. Normalized dose rate to the thyroid from external exposure to ^{137}Cs and to all radionuclides in two regions having predominantly wet and dry deposition.

having low, intermediate, and high reference soil-to-milk transfer factors.

Table 12 contains the ingestion doses received by cases and controls from the ten identified regions (Table 10) of Belarus. There were no cases from region 6 and no controls from region 1. The doses were estimated for a period of 5.7 y, from the time of the accident to the start of the case-control study. For subjects evacuated from the 30-km zone and for some other subjects whose relocations soon after the accident were known, the effects of changes in residence were considered explicitly in the dose estimates. The highest radiocesium ingestion doses are in regions 1–4, where the median doses are in the range 1.3–5.0 mGy. Some individual thyroid doses from radiocesium ingestion exceeded 10 mGy in regions 2 and 3, and individual doses exceeding 1 mGy were estimated for all regions except 8 and 10.

Fig. 11 illustrates the change in the normalized thyroid dose rate due to external exposure to all radionuclides and to ^{137}Cs alone for regions 1 and 8. The normalized dose rate in region 1 is initially higher by about a factor of four than the comparable value for region 8, owing to the larger amounts of radionuclides other than ^{137}Cs (Table 10). The effect of wet deposition (region 8) can be seen in the small difference between the two normalized thyroid dose rates for ^{137}Cs.

Table 13 gives the external exposure contributions to thyroid doses received by cases and controls in the ten identified regions of Belarus during the period of exposure considered for the study. The largest external doses were estimated for the evacuated residents of region 1, which ranged from 8 to 96 mGy, with a median value of 20 mGy. Thyroid doses from external exposure ranging from 0.16 to 92 mGy were estimated for subjects in regions 2 and 3. Only one dose estimate was <1 mGy; however, eight of the individual external dose estimates

Table 12. Thyroid dose estimates (mGy) for ingestion of radiocesium by cases and controls.

Region	Number of cases	Median dose (mGy)	Range of doses (mGy)	Number of controls	Median dose (mGy)	Range of doses (mGy)
1	5	3.3	2.4–3.5	0	–	–
2	27	2.9	0.40–11	21	0.96	0.48–6.4
3[a]	17	5.0	0.67–19	27	2.4	0.50–28
4	19	2.2	1.7–2.2	13	2.1	1.8–2.2
5[a]	9	0.64	0.19–1.4	61	0.30	0.090–3.1
6[a]	0	–	–	8	1.3	1.3–1.3
7	10	0.31	0.30–5.6	16	1.1	0.26–7.6
8[a]	5	0.19	0.19–0.19	20	0.19	0.19–0.19
9	13	0.11	0.070–0.76	44	0.14	0.060–1.2
10	2	0.050	0.050–0.050	4	0.055	0.050–0.060

[a] Region in which wet deposition was predominant.

Table 13. Thyroid dose estimates (mGy) for external exposure of cases and controls.

Region	Number of cases	Median dose (mGy)	Range of doses (mGy)	Number of controls	Median dose (mGy)	Range of doses (mGy)
1	5	20	8–96	0	–	–
2	27	11	2.3–64	21	3.0	1.2–53
3[a]	17	12	1.5–52	27	5.7	0.16–92
4	19	2.7	2.6–2.9	13	2.7	2.6–2.9
5[a]	9	1.2	0.30–2.5	61	0.75	0.18–21
6[a]	0	–	–	8	0.52	0.51–0.54
7	10	0.54	0.54–10	16	2.1	0.99–18
8[a]	5	0.16	0.16–0.16	20	0.16	0.16–0.17
9	13	0.12	0.074–2.2	44	0.093	0.059–3.9
10	2	0.085	0.061–0.11	4	0.061	0.061–0.080

[a] Region in which wet deposition was predominant.

Table 14. Total thyroid dose estimates (mGy) for subjects and contributions of secondary sources of radiation dose.

	Total thyroid dose (mGy)		Percentages of total thyroid dose from sources other than ^{131}I					
			Short-lived iodines[a]		^{137}Cs, ^{134}Cs ingestion		External exposure	
Region	Median	Range	Median	Range	Median	Range	Median	Range
1	2,700	800–4,800	8.4	7.1–8.6	0.13	0.051–0.33	1.1	0.27–2.1
2	590	78–3,000	3.2	2.7–6.0	0.61	0.020–2.1	1.4	0.23–8.6
3	540	3.6–3,300	2.8	2.0–3.2	0.69	0.16–11	1.7	0.12–59
4	230	130–580	1.1	0.59–6.2	0.94	0.31–1.7	1.1	0.49–2.0
5	81	7.6–1,500	0.77	0.58–3.6	0.59	0.08–2.0	1.1	0.23–3.3
6	82	55–190	1.1	0.58–1.4	1.6	0.30–2.3	0.63	0.28–0.92
7	76	21–590	1.7	1.3–1.9	1.1	0.25–3.8	1.9	0.92–4.5
8	33	22–100	0.40	0.30–0.42	0.58	0.15–0.87	0.49	0.17–0.73
9	11	3.1–300	1.7	1.4–2.1	1.1	0.11–4.8	1.2	0.02–5.6
10	5.4	2.5–8.2	0.35	0.31–0.83	0.95	0.66–2.3	1.5	0.81–2.4

[a] Includes ^{132}I, ^{133}I, ^{135}I, and precursor isotopes ^{131m}Te and ^{132}Te.

exceeded 50 mGy in those two regions. The lowest median doses due to external exposure were received by subjects residing in regions 8, 9, and 10.

Table 14 provides a summary of the thyroid doses received by all subjects, by region, and a breakdown of the contributions of the minor sources of thyroid dose. For regions 1–3 the median contributions of the short-lived radioiodine and radiotellurium isotopes exceeded those due to ingestion of ^{137}Cs and ^{134}Cs and those due to external exposure. In regions 1–3 the total contribution of secondary sources of exposure was less than 14% of the total thyroid dose, except for one control born 8 mo after the accident. For that person, external exposure and ingestion of ^{137}Cs and ^{134}Cs were the predominant sources of thyroid dose. In all other regions (4–10), the contributions of the secondary sources of thyroid dose did not exceed 10% of the total thyroid dose estimated for any subject in the study.

CONCLUSION

Calculations of thyroid doses from ingestion of radiocesiums, the most important long-lived radionuclides in the diet, show that these radionuclides did not contribute substantially to the total thyroid doses received by subjects in the study. The median fractional contribution from this source to the total thyroid dose was 0.76%; minimum and maximum contributions were 0.020 and 11%, respectively.

The contributions to thyroid doses from external radiation were generally higher, with a median fractional contribution of 1.2%. Most of the fractional contributions ranged from 0.020 to 8.6%, but for one control subject born well after the accident, external exposure contributed nearly 60% of the total thyroid dose.

Overall, the contributions of radiocesium ingestion and external exposure were lower than those of short-lived radioiodines and radiotelluriums, but in regions 4–10 contributions of the three secondary sources of dose were generally comparable in magnitude. The median fractional contribution of the short-lived radioiodines and radiotelluriums was 1.7%; it ranged from 0.30 to 8.6%. With one exception, noted above, the fraction of the thyroid dose due to external exposure, radiocesium ingestion, and short-lived radioiodines was less than 14%.

REFERENCES

Alexakhin R, Firsakova S, Rauret I, Dalmau G, Arkhipov N, Vandecasteele CM, Ivanov SF, Sanzhakova N. Fluxes of radionuclides in agricultural environments: main results and still unsolved problems. In: The radiological consequences of the Chernobyl accident. Proceedings of the First International Conference, Minsk, Belarus, 18–22 March 1996. Brussels-Luxembourg: European Commission; 1996: 39–47.

Astakhova LN, Anspaugh LR, Beebe GW, Bouville A, Drozdovitch VV, Garber V, Gavrilin YuI, Khrouch VT, Kuvshinnikov AV, Kuzmenkov YuN, Minenko VP, Moschik KV, Nalivko AS, Robbins J, Shemiakina EV, Shinkarev S, Tochitskaya SI, Waclawiw MA. Chernobyl-related thyroid cancer in children of Belarus: a case-control study. Radiat Res 150:349–356; 1998.

Balonov MI, Travnikova IG. Importance of diet and protective actions on internal dose from Cs radionuclides in inhabitants of the Chernobyl region. In: Merwin SE, Balonov MI, eds. The Chernobyl papers. Richland, WA: Research Enterprises Inc.; 1993: 127–166.

Baverstock K. Half-time for clearance of isotopes of caesium in man. In: Age-related factors in radionuclide metabolism and dosimetry. Dodrecht: Martinus Nijhoff Publishers; 1987: 215–220.

Buzulukov YuP, Dobrynin YuL. Release of radionuclides during the Chernobyl accident. In: Merwin SE, Balonov MI, eds. The Chernobyl papers. Richland, WA: Research Enterprises Inc.; 1993: 3–21.

Drozdovitch VV, Minenko VF. Internal exposure from ingestion of foodstuffs contaminated by radiocesium. Part II: Exposure doses to urban population of Belarus. Med Biological Aspects Chernobyl Accident 3:3–8; 1996 (in Russian).

Eckerman KF, Ryman, JC. External exposure to radionuclides in air, water, and soil. Federal Guidance Report No. 12. Washington, DC: US Environmental Protection Agency; EPA 402-R-93-081; 1993.

Fesenko SV, Alexakhin RM, Spiridonov SI, Sanzharova NI. Dynamics of 137Cs concentration in agricultural products in areas of Russia contaminated as a result of the accident at the Chernobyl nuclear power plant. Radiat Prot Dosim 60:155–166; 1995.

Garner RJ. A mathematical analysis of the transfer of fission products to cows' milk. Health Phys 13:205–212; 1967.

Gavrilin Yu, Khrouch V, Shinkarev S, Drozdovitch V, Minenko V, Shemiakina E, Ulanovsky A, Bouville A, Anspaugh L, Voilleque P, Luckyanov N. Individual thyroid dose estimation for a case-control study of Chernobyl-related thyroid cancer among children of Belarus—Part I: 131I, short-lived radioiodines (132I, 133I, and 135I) and short-lived radiotelluriums (131mTe and 132Te). Health Phys 86:565–585; 2004.

International Commission on Radiological Protection. Age-dependent doses to members of the public from intakes of radionuclides: Part I. Oxford: Pergamon Press; ICRP Publication 56; Ann ICRP 20(2); 1989.

Izrael YA, Vakulovsky SM, Vetrov VA, Petrov VN, Rovinsky FI, Stukin ED. Chernobyl: radioactive contamination of the environment. Leningrad: Hydrometeorology Publishing House; 1990 (in Russian).

Jacob P, Rosenbaum H, Petoussi N, Zankl M Calculation of organ doses from environmental gamma rays using human phantoms and Monte Carlo methods. Part II: Radionuclides distributed in the air or deposited on the ground. Munich: GSF-Bericht; 12/90; 1990.

Knatko VA, Mayall A, Drugachenok MA, Matveenko II, Mironov VP. Radiation doses in southern Byelorussia from the inhalation of specific radionuclides following the Chernobyl accident. Radiat Prot Dosim 48:179–183; 1993.

Korneev S, Tretyakevitch S, Minenko V, Ulanovsky A. External exposure. In: Bailiff ID, Stepanenko V, eds. Retrospective dosimetry and dose reconstruction. Experimental collaboration project No 10. Final report. Brussels-Luxembourg: European Community; EUR 16540 EN104-110; 1996.

Lebedev OV, Yakovlev VA. The correlation between 137Cs half-time and age, body mass and height in individuals contaminated from the Chernobyl accident. In: Merwin SE, Balonov MI, eds. The Chernobyl papers. Richland, WA: Research Enterprises Inc.; 1993: 221–243.

Likhtarev IA, Kovgan LN, Vavilov SE, Perevoznikov ON, Litvinets LN, Anspaugh LR, Jacob P, Prohl G. Internal exposure from the ingestion of foods contaminated by 137Cs after the Chernobyl accident—Report 2. Ingestion doses of the rural population of Ukraine up to 12 y after the accident (1986–1997). Health Phys 79:341–357; 2000.

Minenko VF, Drozdovitch VV, Tretiakevitch SS, Ulanovsky AV. Exposure of Belarus population after Chernobyl accident: collective doses and forecast of stochastic effects. Med Biological Aspects Chernobyl Accident 4:50–65; 1996 (in Russian).

Orlov MYu, Snykov VP, Khvalevsky YuA, Teslenko VP, Korenev AI. Radioactive contamination of Russia and Belarus territory after the Chernobyl accident. Atomnaya Energiya 72:371–376; 1992 (in Russian).

Petriaev EP. Estimation of short-lived radionuclides by the relation to long-lived isoptoes at the time of the Chernobyl NPP accident. Final report. Minsk: Belorussian State University; Reg. No. 317/17; 1994 (in Russian).

Savkin MI, Titov AV, Lebedev AN. Distribution of individual and collective exposure doses for the population of Belarus in the first year after the Chernobyl accident. Radiat Risk 7:87–113; 1996 (in Russian).

State Committee on Hydrometeorology of Republic of Belarus. Contamination of the territory of the settlements of Republic of Belarus. Minsk: State Committee on Hydrometeorology; 1994 (in Russian).

United Nations Scientific Committee on the Effects of Atomic Radiation. Sources, effects and risks of ionizing radiation. 1988 Report to the General Assembly with Annexes. New York: United Nations; 1988.

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